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One-hundred years after shrub encroachment: Policy directions towards sustainable rangeland-use

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ABSTRACT

In many shrub-encroached lands, livestock grazing is the dominant land-use type which shapes stakeholders' perspective on single ecosystem service provision (i.e., forage production). Although recent ecological studies suggested otherwise, recognition of multiple ecosystem services from shrublands is rarely translated into policy, likely due to the lack of robust scientific evidence on trade-offs between ecosystem services following shrub removal. Based on meta-analysis from global publications, we found that while shrub removal increased forage provision, such effect was generally short-lived (~5 years). At the same time, shrub removal also increased bare soil, decreased soil nutrients and soil organic carbon, which potentially reduced erosion control and nutrient cycling service. These trade-offs tended to be more prominent with increasing disturbance intensity (i.e., higher shrub removal frequencies or switching from single to multiple shrub removal methods). To encourage adaptation to the provisioning of multiple ecosystem services from shrublands, we provided a framework on how to value such landscapes, including the estimated monetary values that could be generated from maintaining them. Since there will be a time-lag until most monetary values appear, initial incentives may be necessary to encourage the adoption of conservation practices, in addition to efforts to increase society awareness (e.g., ecolabelling, alternative food networks or social media) on multiple ecological benefits from shrublands and investment to provide capacity building for pastoralists.

1. Introduction

Drylands comprise of approximately 40% the Earth's land surface and have considerable, multi-dimensional provisions of ecosystem services (Maestre et al., 2017). Yet, given livestock grazing is the most common land-use in the region (Maestre et al., 2017), traditional perspective of the majority of stakeholders (i.e., local resource users or pastoralists, politicians, non-governmental organizations and scientists) still recognizes rangeland as a provider of a single ecosystem service (i.e., forage provisioning). Accordingly, it is unsurprising if changes in plant communities from grassland to shrubland (i.e., shrub encroachment), which have been occurring over the last 100 years, remain a concern in areas primarily used for grazing.

Although many recent ecological studies have highlighted multiple ecosystem services from shrub-encroached areas (e.g., carbon or C sequestration, biodiversity, nutrient and water cycling, habitat and conservation service) (Eldridge et al., 2011; Archer and Predick, 2014; Eldridge and Soliveres, 2014; Soliveres et al., 2014; Reed et al., 2015; Archer et al., 2017; Wilcox et al., 2017), shrub removal, that has been aggressively applied since 1940s (Archer et al., 2017), is still applied in many dryland regions such as Namibia (Haussmann et al., 2016), South Africa (te Beest et al., 2012) and United States (Wonkka et al., 2016). Non-linear changes in ecosystem functionality with aridity and grazing pressure (Anadón et al., 2014; Eldridge and Soliveres, 2014; Soliveres et al., 2014), which resulted in inconsistent effects of shrub removal practices, could contribute to the difficulties in developing policies for sustainable rangeland use. Concerns that fire causes soil erosion, loss of soil fertility and biodiversity, for example, have led to fire suppression policy despite their historical use in Uganda's savanna (Byakagaba et al., 2018). In contrast, the integration of pastoralist knowledge to control shrubs using prescribed fire has been promoted in Namibia (Sheuyange et al., 2005).

Because increasing evidence showing that areas experiencing shrub encroachment maintain multiple ecosystem service provisions,

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adaptation to shrub encroachment should instead be considered and translated into policies. Such policies will be greatly favored if we understand the trade-offs between different ecosystem services associated with different types of shrub removal methods, including the combined methods. Currently such knowledge remains absent from the literature (Archer and Predick, 2014). This knowledge gap has also been highlighted by Eldridge and Soliveres (2014) and Anadón et al. (2014) who stated that there has been little rigorous, quantitative assessment on the long-term effectiveness of shrub removal, including whether such removal achieves the desirable outcome (i.e., increasing grass productivity) and improves ecosystem provision (e.g., water supply) (Wilcox et al., 2017). To address this knowledge gap and to provide a robust scientific evidence on ecological processes following shrub removal, we conducted a meta-analysis study using data from global publications. We examined some of the most commonly measured ecosystem properties after shrub control practices across different climatic, soil or management conditions to provide an overview for the policy makers on the effects of different shrub control practices on multiple ecosystem services. These analyses, building on the earlier examination on the use of fire to manage shrublands (Daryanto et al., 2019), can provide better understanding on the potential trade-offs between different ecosystem services, which are greatly influenced by ecological processes following different management practices.

More importantly, the lack of integrated socio-economic framework to value different ecosystem services also needs to be addressed (Archer and Predick, 2014; Daryanto et al., 2019) because the driving force to conduct shrub removal is mostly economical. In this perspective, we provided a framework to value different ecosystem services from shrublands and the estimated monetary values for each of those services from different locations in Ethiopia. The region was selected because it has a wide array of ecosystem service provisions from shrublands, including both the use and non-use values (Balana et al., 2012; van Zyl, 2015).

2. Environmental responses and ecosystem services potentially affected by shrub removal

There are three common shrub removal methods (fire, chemical and mechanical) used to manage shrublands that were collected as the basis of our perspective. Details of the database collection and methodology can be found in the e-component of this article. We did not consider biological control method (i.e., insect) because there have been stringent policies for releasing biocontrol agents. Changes in ecosystem properties due to biological control are also expected to occur at a much slower rate compared to other shrub control methods considering the time required for biocontrol agents to weaken or kill the shrubs.

2.1. Forage provisioning service

In general, removal of shrubs managed to increase herbaceous productivity (Fig. 1). However, such effects were usually short-lived (Fig. 1), as also indicated by the common use of multiple or repeating treatments in areas experiencing shrub encroachment. Grass cover generally declines by the time shrub cover increases due to resource competition (Ansley et al., 2004; Haussmann et al., 2016) or reduction in summer precipitation (Daryanto and Eldridge, 2010). In some rare cases, however, low initial shrub cover and density and long-term exclusion from grazing can result in a lasting effect of shrub control (~13 years) on perennial grass productivity (Bates et al., 2005; Pierson et al., 2007; Bates et al., 2017).

2.2. Soil-related ecosystem services

In general, there could be decreasing soil erosion control service with increasing bare soil and decreasing litter cover, even with only a single fire treatment (Fig. 2a). Because burning reduces the resistance of soil to shear, the risk of soil erosion in burnt land with heavy grazing (0.35 livestock unit ha^{-1} yr^{-1}) increases by four-times compared to the unburnt land (Stavi et al., 2017). In areas with low annual rainfall (200–300 mm), even when the stocking rate is quartered, there is still a higher risk of erosion in burnt than unburnt landscape (Stavi et al., 2017). Unless understory cover is maintained following shrub removal, soil erosion could be exacerbated due to the loss of biological crust cover (BCC; Fig. 2a), which has been known to be susceptible to disturbance, such as mechanical (Daryanto and Eldridge, 2010) and chemical shrub removal (Brock et al., 2014). While direct physical disturbance during shrub removal would have been responsible to an almost complete loss of BCC (Fig. 2b), trampling by livestock would compromise its recovery (Daryanto and Eldridge, 2010). Long-term nutrient cycling could also be at risk with depleting litter cover (Fig. 2b).

Our synthesis showed that single shrub removal treatment did not change the organic carbon content in the upper soil layer compared to the untreated control (Fig. 2a–b). The use of low to moderate intensity fire in most fire experiments (i.e., between 1400 and 2100 kW m^{-1}) (Zheng et al., 2016) could be attributed to such results (Fig. 2a). Long-term changes in soil organic carbon (SOC) with mechanical shrub removal also seemed to be insignificant (Fig. 2b), possibly because woody or lignified litter serves as long-term C storage (Daryanto et al., 2012). In the absence of repetitive soil disturbance, belowground C can quickly accrue in the order of 0.15-0.35 t ha^{-1} $year^{-1}$ (Daryanto et al., 2013; Derner et al., 2014), despite initial application of mechanical or chemical shrub removal.

Meanwhile, the landscape-scale effects of shrub removal on greenhouse gas (GHG) mitigation remain to be elucidated. So far, changes in SOC and shrub encroachment show non-linear relationship, depending

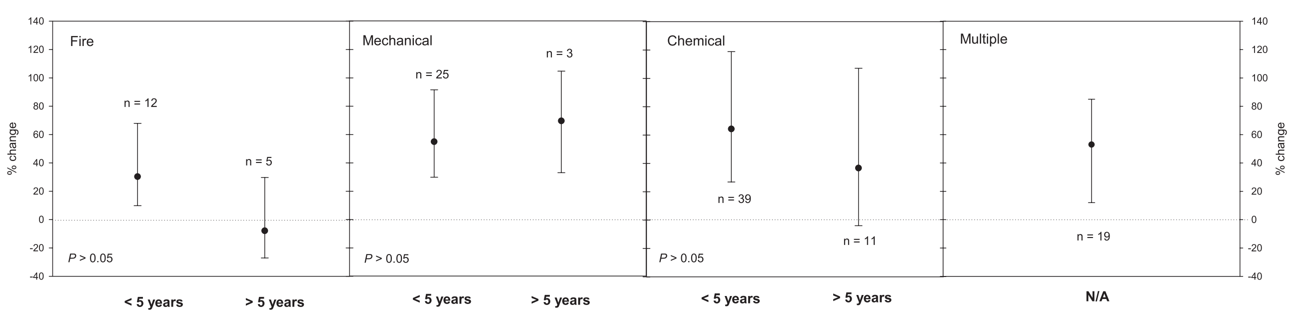

Fig. 1. Changes in herbaceous biomass after single treatment or multiple treatments (N/A means duration could not be determined for multiple treatments). Black dots represent the mean of *lnR* with error bar representing the 95% confidence interval (CI). A negative value indicates a reduction due to shrub removal treatment in comparison to untreated control which is only statistically significant when the CI does not overlap zero. Letter ‘n’ indicates the number of samples. The *P* values indicate statistical significance between short (< 5 years) and long-term (> 5 years).

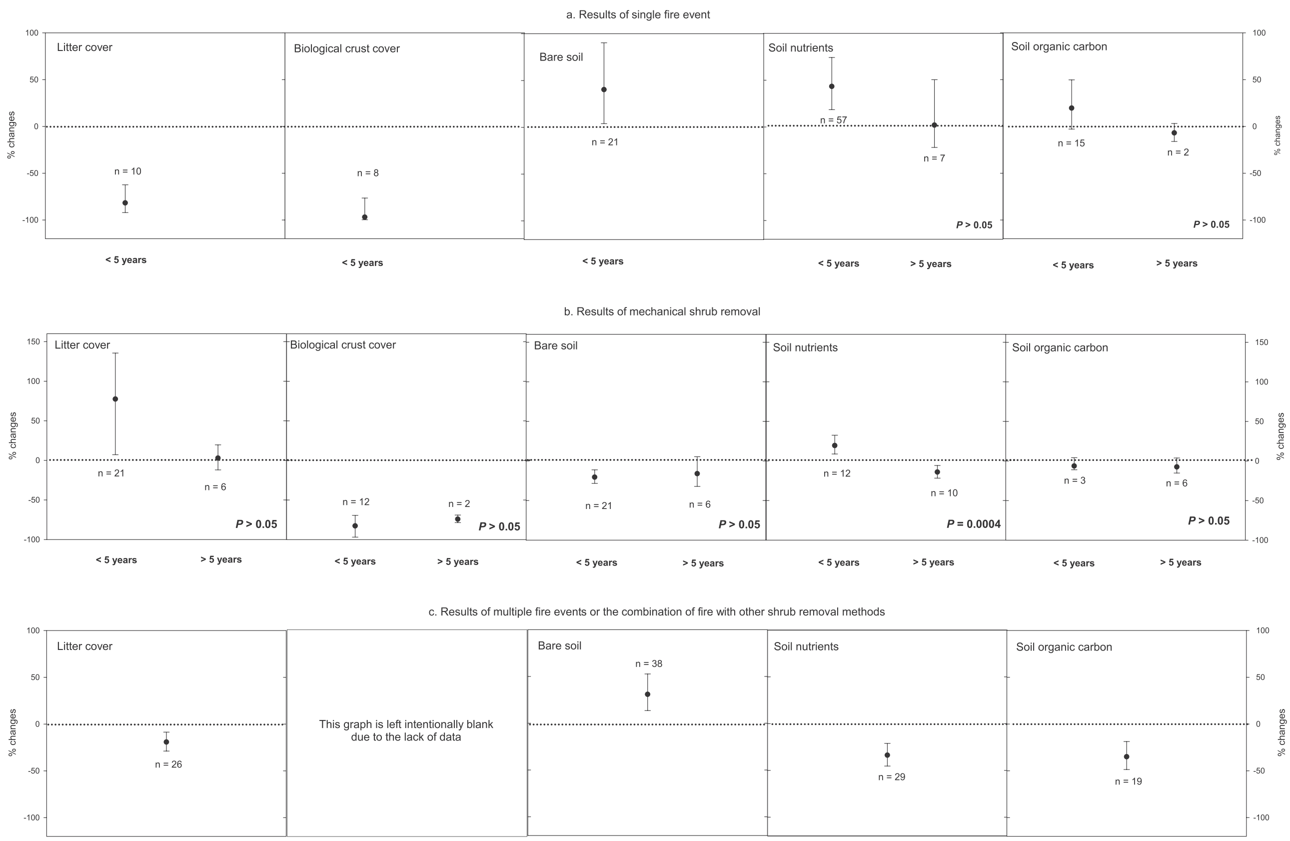

Fig. 2. Changes in soil and landscape characteristics after single fire (a), after mechanical shrub removal (b) and after multiple fire events or a fire with other shrub removal methods (c). Note differences in y-axis and x-axis between panel a, b and c. No data were available for litter cover, biological soil crust cover and bare soil after single fire event for > 5 year-duration. Black dots represent the mean of *lnR* with error bar representing the 95% confidence interval (CI). A negative value indicates a reduction due to shrub removal treatment in comparison to untreated control which is only statistically significant when the CI does not overlap zero. Letter 'n' indicates the number of samples. The *P* values indicate statistical significance between short (< 5 years) and long-term (> 5 years).

on mean annual precipitation and corresponding shrub cover (Knapp et al., 2008; Eldridge and Soliveres, 2014; Soliveres et al., 2014). Increasing frequency and intensity of disturbance may be detrimental to plant regeneration capacity and SOC, leading to soil erosion and degradation. Such decline in soil quality can significantly reduce the CH_4-utilizing bacteria population and soil CH_4 oxidation capacity (Castaldi and Fierro, 2005). Burnt soils also enhance the emission of: (i) nitrous oxide (N_2O) with increasing availability of ammonia (NH_4^+) (Fierro and Castaldi, 2011) as well as (ii) carbon dioxide (CO_2) due to increasing soil respiration (Muñoz-Rojas et al., 2016).

2.3. Biodiversity and habitat conservation services

In terms of biodiversity and habitat conservation service, shrub removal might favor different species as exemplified by the finding of Daryanto and Eldridge (2010) who found that ploughed areas tend to be dominated by shrubs that can regenerate by previously dormant (epicormics) buds or root suckering, while burnt areas tend to be dominated by fire-tolerant species (Masocha et al., 2011). Similarly, shrub removal also produces homogenous young shrubs which are unlikely to provide habitat or any other biodiversity conservation service. Even with possible increases in herbaceous diversity due to larger canopy opening and/or more available soil moisture, the composition of herbaceous community may shift towards unfavourable species. Findings from multiple studies indicated that burnt areas tend to be more vulnerable to invasion by exotic and annual species invasion (Steers and Allen, 2010; Masocha et al., 2011; Miller et al., 2014). A profound example is the expansion of cheatgrass (*Bromus tectorum*) in cold American desert regions which can reach up to 10-times higher than unburnt control (Miller et al., 2014).

Far less known are the impacts of shrub removal, particularly the use of herbicides, on microbial diversity such as BCCs. So far, our understanding has been restricted by the diversity of species and herbicide used. Initial evidence, however, suggested that BCCs, particularly cyanobacteria and soil algae, are adversely impacted by simazine, a photosynthesis inhibitor (Zaady et al., 2004). Glyphosate and imazapic also reduce the number of moss and lichen since ESPS (5-enolpyruvylshikimate-3-phosphate) and ALS (acetolactate synthase) enzymes that become the target of these herbicides are also found in microbes. Meanwhile the effect of picloram, a synthetic auxin, is not as obvious (von Reis, 2014).

2.4. Other ecosystem services

There are also potential trade-offs with respect to other ecosystem services following shrub removal. Socio-cultural values are often neglected, despite findings that shrub-encroached areas may be able to support wildlife and provide recreational benefits for tourists (Reed et al., 2015; van Zyl, 2015). Although increases in woody cover correspond to increasing transpiration relative to the overall evapotranspiration (Wang et al., 2014), removal of woody plants, in most cases, does not result in increasing water supply (Wilcox et al., 2017). Reduction in infiltration capacity with increasing bare soil might be a reason (Eldridge et al., 2015), in addition to increasing soil hydrophobicity, at least in the short-term, with fire treatment (Wilcox et al., 2017). Increasing water supply service with shrub removal only occurs in areas having annual precipitation above 500 mm as winter precipitation, as well as deep and permeable (sandy) soils (Wilcox et al.,

2017). Yet other ecosystem properties (e.g., herbaceous cover and diversity, bare soil and litter cover) were less influenced by soil texture (e-component Fig. S1), most likely because of interactions with grazing or the indirect relationships between, for example, bare soil and soil texture. Herbaceous productivity is instead favored by fine-textured soils, which retain water and nutrients near the surface (Archer et al., 2017).

Overall, shrub removal practice could be beneficial when short-term forage provisioning service was the only service considered. But when other ecosystem services were included, the net effects of shrub removal practice could be negative considering the trade-offs between forage provisioning and other ecosystem services. These results were almost similar regardless of shrub removal methods (Fig. 2a & b), likely because most shrub removal treatments are indiscriminative (e-component Table S1); they do not differentiate between shrub cover, grazing pressure, shrub composition, edaphic or climatic variability. There was, however, a tendency that increasing disturbance intensity (i.e., from single to repeating treatments) and/or rainfall could lead to greater trade-offs between forage provisioning and other ecosystem services (Fig. 2c; e-component Fig. S2). Increasing impacts of shrub removal *per se* and increasing stocking rate with rainfall could be attributed to these trade-offs, particularly with regards to erosion control and nutrient cycling (Fig. 2c). Severe erosion risk, for example, is recorded with multiple fire events due to increasing runoff coefficient and decreasing saltation threshold (Hosseini et al., 2016). Land degradation and its associated loss of ecosystem services due to, among others, soil erosion, costs the world an estimated $6.3 trillion annually (~8.3% of global gross domestic product in 2016) (Ding et al., 2017). Therefore, adaptation to shrub encroachment by recognizing the multiple ecosystem services (i.e., regulatory, cultural, and supporting) is important to ensure sustainable land-use in shrub-encroached lands (Reed et al., 2015).

3. Integrated policy direction for shrub-encroached lands

Although different ecosystem services from shrub-encroached lands have been recognized in many ecological studies (Archer et al., 2017; Eldridge et al., 2011; Reed et al., 2015; Wilcox et al., 2017), the integration of these scientific findings into policies has taken a slower pace likely because ecologists failed to recognize the economic reasons behind shrub removal. In human-dominated ecosystems, in which shrub encroachment is seen as a threat to the sole source of income, ecological principles alone are inadequate to sustainably manage the environment. As shrub encroachment is expected to remain with climate change and efforts to combat shrub encroachment for the sole purpose of grazing can be considered putative (Archer et al., 2017), policies that facilitate adaptation to the phenomenon, including from the social and economic perspective should instead become a more feasible option (Fig. 3). Consideration of those services would lead to a long-term protection of the landscape and to provide a broader range of environmental and social benefits (Reed et al., 2015), equivalent to an estimated 7–30 times return with every unit of currency invested for such effort (Ding et al., 2017).

To facilitate an integrated ecological, economic and social policy development, we identified several gaps between science and policy that need to be developed. The first one is research on trade-offs or synergies between different ecosystem services, including cultural services based on simultaneous responses and their management (Fig. 3), considering that interactions (trade-offs and synergies) exist between different ecosystem services (Raudsepp-Hearne et al., 2010). Currently our understanding is limited because the existing knowledge was primarily derived from studies that have examined a single service or a subset of services in the ecosystem. Accordingly, such quantification also needs constant monitoring, standardized methods and parameters for assessing landscape conditions (and ecosystem services) that are easy to be conducted even by pastoralists for adaptive management options (Tongway and Hindley, 2000; Reed and Dougill, 2010). Given that the spatial distribution of grazers and societal feedbacks (e.g., mobile vs sedentary livestock grazing, customary land rights) could also affect the interactions between ecosystem services (Raudsepp-Hearne et al., 2010), field studies need to be supported by remote sensing and modelling to take advantage of their long-term and broad-scale nature and to better understand the complex interacting factors (i.e., between environment, humans, economic and social drivers) because management can alter both the target and non-target processes (Figs. 1 and 2).

Understanding of the aforementioned ecosystem services then needs to be supported by detailed economic analysis to value each service, including those that may require longer term to take effect (Ding et al., 2017). Since provision of ecosystem services varies greatly from one location to another as well as from time to time, in this perspective, we provided a valuation framework to calculate the monetary values for different ecosystem services from shrublands and the estimated monetary values for each of the services, including the regulation, habitat and cultural functions (Fig. 4; Table 1). The framework could also be used as the basis to reward conservation practices using incentives because most monetary values could only be realized after certain time lag. Production of fuelwood that comes alongside with forage production, for example, only stabilizes after 10 years of land protection, before it can generate USD ~36 ha^{-1} $year^{-1}$ in Tigray, Ethiopia. Other ecosystem services, such as increased infiltration and erosion control, also depend on the age after restoration (Table 1). Further analysis, however, will be necessary to identify who benefits and who pays the cost, considering the partitioning of different types of ecosystem services are not the same. Erosion prevention service, for example, is shared in a greater extent by areas closer to the vicinity of the encroached lands, while GHG mitigation service through C sequestration is shared by a much larger community (i.e., the entire world).

Since different communities may have different values and appreciation towards different ecosystem services, priorities of ecosystem services should be given to those that reflect stakeholders' concerns and stewardship goals. Wunder (2005) provided a systematic approach on how to design a valuation or payment for ecosystem services (PES) and examples of successful community- or government-initiated PES have been discussed elsewhere (Zhen et al., 2014; Bhatta et al., 2018). Most importantly, we need to understand the 'willingness-to-pay'; a maximum amount that pastoralists are willing to limit, for example their stocking rate or to shift their animal enterprise, in return for the desired services (e.g., improved soil fertility, soil erosion, etc.). Review by Hacker and Alemseged (2014) showed that shifting animal enterprise (e.g., from sheep to goat) can result in less bare ground, soil erosion and weed invasion due to goats' flexible diet. At the same time, other stakeholders also need to be asked questions such as: 'If pastoralists reduce their stocking rate to conserve grasslands and their financial losses should be compensated, how much you are willing to pay for the economic loss/conservation to meet their end'? Such policies are therefore more likely to be successful if an organized environment to integrate stakeholders' opinion is developed (Basupi et al., 2017). Increasing evidence from multiple continents suggests the importance of integrating stakeholders' opinion into policy development. Examples of such action have taken place in Botswana with the involvement of pastoralists in participatory mapping to better manage resources in the dry area (Basupi et al., 2017), recognition about the importance of traditional practices such as stock mobility in Uganda (Byakagaba et al., 2018), and Australia following Aboriginal values (McAllister, 2012).

By removing price constraints that can deter some pastoralists from selling sustainably grown livestock at affordable price, incentives may reduce the gap between social classes. Incentives can also be translated in forms of knowledge dissemination to improve human capital because government can adjust the compensation methods (e.g., compensation via appropriate technologies and knowledge), including marketing advantage (e.g., eco-labelling) for products grown under a more sustainable method relative to conventional production practices. Currently the tools for promoting society awareness, behaviour change

Fig. 3. Integrated framework from social, economic and ecological perspective for sustainable rangeland management.

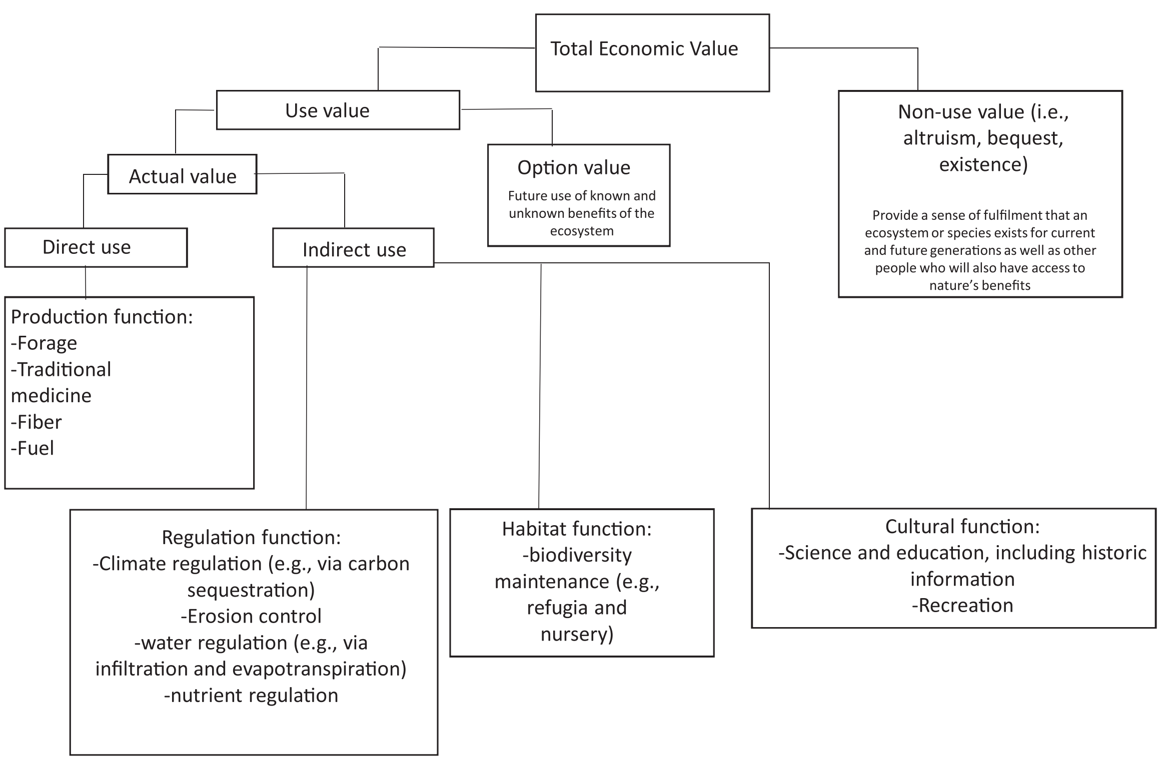

Fig. 4. A framework for ecosystem service valuation in shrub-encroached lands (adapted from Department of Environment, Food and Rural Affairs, United Kingdom, 2007) and The Economics of Ecosystems and Biodiversity (TEEB, 2010).

Table 1
Common methods of valuation for different ecosystem services from shrub encroached lands (adapted from Department of Environment, Food and Rural Affairs, United Kingdom, 2007 and TEEB, 2010) and examples of estimated services and values that can be obtained from protecting degraded areas in different locations in Ethiopia. Services and values are obtained from Mekuria et al. (2011); Balana et al. (2012); Government of the Federal Democratic Republic of Ethiopia (2014); van Zyl (2015), and Ministry of Environment, Forest, and Climate Change, Federal Democratic Republic of Ethiopia (2017).

Type of ecosystem service	Type of economic value	Common valuation method	Observable service	Estimated value*
Production service (e.g., forage, fuel, fiber)	Direct use	Market price	Fuelwood production: ranging from 0 (year 0–5) to a constant of ~0.80 t ha^{-1} after year 10 Forage production: ranging from 2.85 (year 5) to a constant of ~1.01 t ha^{-1} after year 10 Traditional food source: *Moringa stenopelata, Cordia africana, Balanites aegyptiata, Dovyalis abyssinica, Carissa edulis, Rosa abyssinica*	Fuel wood: USD 30 t^{-1} (Balana et al., 2012) Forage: USD 12 t^{-1} (Balana et al., 2012)
Regulation service (e.g., nutrient cycling, infiltration, pollination, carbon sequestration)	Indirect use	Production function approach, avoidance cost, contingent valuation	Erosion control: ~39 m^3 ha^{-1} (mass of sediment multiplied by soil bulk density) Vegetation belt: 477-715 km (length of the exclosure) Increased infiltration: 40-130% of rainfall (depending on the presence of run-on and duration after restoration) Pollination service for wild coffee plants Carbon sequestration: ranging from 67.3 (year 5) to 102.5 t ha^{-1} (25 years)	Reservoir protection from sedimentation: USD ~8.2 ha^{-1} $year^{-1}$, equivalent to the volume of trapped sediment multiplied by the digging cost of reservoir (USD 0.21 m^{-3}) (Balana et al., 2012) Crop field protection from flood: USD ~4.3 ha^{-1}, equivalent to 50-100% reduction of crop damage valued at USD 68.9 ha^{-1} (Balana et al., 2012) Available water for irrigation after groundwater recharge: ranged from USD 7.9 to USD 12 ha^{-1} depending on duration after restoration. Value is estimated from the incremental increase of maize yield productivity (Balana et al., 2012) Pollination value of restored land: USD 55 ha^{-1} $year^{-1}$ (vs USD 10 ha^{-1} $year^{-1}$ for degraded landscape) (van Zyl, 2015) Ecosystem C storage: USD 992 ha^{-1} (Mekuria et al., 2011)
Habitat and supporting service (e.g., biodiversity conservation)	Indirect use	Market price, including income, contingent valuation	Provision of habitat for ~1000 endemic woody plant species, some of those have high economic values including different frankincense, myrrh and other gum producing shrubs such as *Boswellia papyrifera*, *B. neglecta*, *B. rivae*, *B. microphylla*, *B. ogadensis*, *Commiphora myrrha*, *C. guidot*, *C. erythraea* or *C. africana*, *Acacia senegal*, *A. seyal* (average production = 71 × 10^3 t $year^{-1}$), forest coffee (average production = 343 × 10^6 t $year^{-1}$), and wild spices (up to 1.2 × 10^3 t $year^{-1}$) (Government of the Federal Democratic Republic of Ethiopia, 2014)	Export from natural gum and resin: up to USD 12 × 10^6 $year^{-1}$ Forest coffee: USD 7.5 kg^{-1} Wild spices: USD 2 kg^{-1} (Ministry of Environment, Forest, and Climate Change, Federal Democratic Republic of Ethiopia, 2017)
Cultural service (e.g., recreation, science, history)	Indirect use	Hedonic pricing, travel cost	Preservation of traditional medicinal practices using different shrub species such as *Calendula ofcinalia* for haemorroids, *Eucalyptus globulus* for muscular problem, *Matricaria chamomilla* for headache, *Datura stramonium* for asthma, etc, with an average use of 56 × 10^3 t $year^{-1}$ Protected areas ~83,700 ha in 2014	Average pharmaceutical value: USD 4.4 ha^{-1} (Ministry of Environment, Forest, and Climate Change, Federal Democratic Republic of Ethiopia, 2017) Ecotourism: USD ~2 × 10^9 in 2012 (van Zyl, 2015)
All ecosystem services	Option and non-use value	Contingent valuation, choice modelling	Examples are the existence value of a land (and its associated iconic species that become the national identity), the amount of projected harvest and carbon sequestered during certain time period	Existence value, water supply service, pollination and pest control service of a degraded semi-arid landscape increase from only USD 1.2 to USD 12, USD 256 to USD 1,024 and USD 232 to USD 1,449 respectively when it is restored within the next 20 years (van Zyl, 2015)

* Based on 2007 exchange rate (1 USD = 10 ETB).

and trust building from pastoralists regarding conservation practices and ecosystem services from shrublands appear inadequate and require more than simply a scientific ecological proof. Approaches through alternative food networks or social media can be a powerful strategy to strengthen the ethical belief and action, and even to transform individual attitudes into collective statements (Klinglmayr et al., 2017). Using social media can raise public awareness and greater interaction with producers, leading to greater trust in product quality (Gummerus et al., 2017), which may in turn become a driving force for pastoralists to change their management practices. In addition, government willingness to provide capacity building (e.g., training and technical support) to diversify income sources beyond livestock grazing is essential, starting from the most straightforward (e.g., the use of shrubs as materials for medicine or charcoal) (Reed et al., 2015) to the more sophisticated PES with C sequestration or C trading (Daryanto et al., 2013). The government can invite multilateral banks, philanthropic organizations or even society to invest on this capacity building, by offering mechanisms that reduce risks such as insurance or tax credit (Ding et al., 2017).

4. Conclusions

Recognition and valuation towards multiple rather than single ecosystem service from areas affected by shrub-encroachment appear to be a promising strategy for sustainability. Yet this recognition has not yet so far been converted into policy, likely because a robust scientific evidence is still critically required to examine trade-offs or synergies between different ecosystem services, including detailed economic analysis to support the valuation of shrub-encroached lands. This step is crucial because the driving force to conduct shrub removal is mostly economical with reduction in grazing capacity threatening the sole source of income. Therefore, the practice only considered forage provisioning service. The lack of understanding about the social and economic drivers of degradation often hinders the efforts to ecological recovery. With greater recognitions among stakeholders on the importance of supporting services beyond forage production in the rangelands, opportunity to translate appraisal of multiple ecosystem services into policy is wide open, but it must be simultaneously supported by increasing society's awareness and capacity building for pastoralists to diversify income sources. Adaptive ecological, economic and social capability which provides learning opportunities for all stakeholders represents a central component to develop more effective grass-root policies.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.landusepol.2019.03.008.

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